

Effects of Nutrient Restrictions on Confined Animal Facilities: Insights from a Structural-Dynamic Model

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Nutrient emissions from animal feeding operations continue to degrade water and air quality. New regulations will limit the amounts of nutrients that can be locally applied to land. In this paper, a structural-dynamic model of a livestock-crop operation is calibrated with data from a representative farm and is used to predict the effects of nitrogen regulations. Policy simulations clarify the importance of dynamic elements and demonstrate three main results: (1) cost estimates for large producers are higher than suggested by previous studies; (2) cross-media pollution effects are potentially significant; and (3) improved input management appears most promising for reducing both water and air emissions and waste management costs. Implications for policy and future research are discussed.

Les émissions d'éléments nutritifs provenant des exploitations d'élevage continuent de dégrader la qualité de l'eau et de l'air. De nouveaux règlements limiteront les quantités d'éléments nutritifs qui pourront être appliqués localement sur les terres. Dans le présent article, un modèle structurel dynamique d'exploitation mixte (élevage-culture) a été étalonné d'après des données tirées d'une exploitation typique et a été utilisé pour prédire les répercussions des règlements sur la gestion de l'azote. Des simulations de politiques clarifient l'importance des éléments dynamiques et démontrent trois résultats principaux: 1. l'évaluation des coûts dans le cas des grosses exploitations est plus élevée que celle déterminée dans des études antérieures; 2. les effets de la pollution touchant plusieurs milieux sont potentiellement importants; 3. une meilleure gestion des intrants semble plus prometteuse pour réduire les émissions dans l'eau et l'air et pour réduire les coûts de gestion des déchets. Nous traitons des implications pour la politique et la recherche future.

INTRODUCTION

Over the past 25 years global livestock production has nearly doubled with a trend toward larger and more concentrated operations (FAO 2007). In lock-step with this trend are increases in the waste byproducts from these operations, particularly excess nutrients (Gollehon et al 2001). Given the potentially negative environmental and health impacts associated with nutrient pollution, animal feeding operations (AFOs), and their byproducts have attracted the attention of regulatory agencies and environmental initiatives worldwide (Shortle et al 2001; Criss and Davidson 2004). Much of this attention has focused on reducing impacts to water quality. The European Union's Nitrate Directive

requires member states to identify Nitrate Vulnerable Zones and produce action plans that target manure waste applications from animal operations (Latacz-Lohmann and Hodge 2003). In New Zealand, concern over nitrate levels in surface and ground water also has prompted the government to require that dairy farmers implement land-based effluent disposal systems (Cassells and Meister 2001). The Canadian province of Alberta historically has regulated manure application rates based on nitrogen content but is considering implementing stricter phosphorus-based standards (Smith et al 2006).

In the United States, the largest contributor to lake degradation and third largest contributor to river and stream degradation is nutrient pollution, primarily from agriculture (Shortle et al 2001). Nitrate contamination of ground water also is a concern. Nationally, approximately 22% of domestic wells in agricultural regions exceed the federal maximum contaminant level for nitrate (Ward et al 2005). Previous studies have left little doubt that large AFOs are significant contributors to these problems (Lowry 1987; Mackay and Smith 1990; Harter et al 2002), especially given the ongoing trend toward consolidation (Meyer 2000; Shortle et al 2001). In response to these water quality problems, the Clean Water Act was revised in 2003 to regulate large-scale animal operators in a manner similar to their European, New Zealand, and Canadian counterparts. Previously, the main focus of the Clean Water Act with regard to large-scale concentrated animal feeding operations (CAFOs) was on restricting the discharge of waste, either directly or through a conveyance system, into water. Under the new amendments, all CAFOs will be required to implement a nutrient management plan (NMP) whereby the application rates for manure must be consistent with agronomic rates of nutrient uptake by crops and application must be done in a manner that minimizes nitrogen and phosphorus runoff into surface waters (Federal Register 2003).¹

However, as noted by Aillery et al (2005; p.1), "a logical response by producers operating under a nitrogen-based plan might be to reduce the nitrogen content of manure spread on fields by enabling nitrogen to volatilize into the atmosphere from uncovered lagoons or by applying animal waste to land without incorporating it into the soil." Failure to appreciate the potential response by livestock operators to more stringent water quality regulations and the possible impacts of this response on air quality could lead to costly future regulatory adjustments and/or violations of other environmental standards. Such cross-media pollution concerns are legitimized by the fact that animal manure is responsible for 33% of all human-related nitrous oxide emissions and 50% of all terrestrial ammonia emissions, both of which contribute to particulate matter air pollution and global warming (NRC 2002). Furthermore, ammonia emissions in rural areas in the United States are approaching levels that might trigger federal action under the Clean Air Act requiring states to regulate these emissions (Ribaud and Weinberg 2005). Although some state-level agencies (e.g., the San Joaquin Valley Air Pollution Control District) are pursuing more effective air regulations, these efforts remain uncoordinated with changes to water quality regulations.

A key issue raised by these ongoing shifts in environmental regulations is the potential for negative economic impacts on AFOs. Significant policy-induced increases in production costs, possibly resulting from herd reductions, reduced crop production, and/or increased waste management costs, ideally should be weighed against the benefits of reduced pollution. Substantial cost increases also could make policy implementation

difficult in some regions due to equity concerns or could induce unanticipated changes in the industry (e.g., restructuring, relocation). Several recent studies have sought to estimate these cost impacts by utilizing farm-level optimization models (e.g., Ribaudo et al 2003; Ribaudo et al 2004; Aillery et al 2005; Huang et al 2005; Ribaudo and Agapoff 2005). Collectively, these studies present a fairly broad range of possible economic impacts, with much of the variability attributable to the type of AFO considered (dairy, swine, or poultry), its size and characteristics (e.g., type of manure handling system), the type of NMP (nitrogen or phosphorus), and the amount of off-farm land available for applying manure. Two other studies (Feinerman et al 2004; Kaplan et al 2004) utilize regional economic models to derive aggregate production changes and welfare losses.

The focus of this article is on farm-level modeling of nitrogen-based NMP implementation and the associated environmental and economic impacts. We extend the existing literature in several ways. First, unlike previous studies, which utilize static (single period) frameworks, we implement a structural-dynamic optimization model that enables us to address temporal aspects of the problem.² This approach, which intuitively provides a more realistic model of a farm, also allows us to account for additional constraints on operators that may result in higher compliance costs and longer transition periods before pollution reduction goals are achieved. Second, we calibrate our model to a relatively large AFO. Previous studies typically define “large” farms as having at least 700 animals, thus qualifying as a CAFO. Our model farm is about three times this size and therefore indicative of future conditions if consolidation continues as expected. Third, rather than limit our economic analysis to a partial farm model focusing on waste disposal decisions, we implement a whole-farm model that also includes components for herd management and crop production. And fourth, instead of limiting our assessment of environmental impacts to nutrient application rates, we also model nitrate leaching and ammonia volatilization. This allows us to incorporate two important additional aspects of farm management into our model: nonuniform irrigation and evaporation ponds.

Our specific objectives are threefold: (1) estimate NMP implementation costs with and without additional air regulations; (2) estimate NMP-induced environmental impacts with and without additional air regulations, particularly the time required for pollution reductions to be achieved and the potential for cross-media effects; and (3) evaluate whether our additional model detail and complexity produce significantly different results compared to previous studies. The paper is organized as follows. First, we develop a structural-dynamic model of a large AFO that represents a modern dairy farm in California’s San Joaquin Valley.³ Next we determine the preregulation steady-state operating position for the farm and verify that it is well-calibrated with our study site. Then we simulate the optimal management response to several scenarios: (1) agronomic restrictions on nitrogen application rates, (2) agronomic restrictions with adoption of improved input management techniques, (3) agronomic restrictions with uniform irrigation, and (4) agronomic restrictions with selective culling of the herd. We then evaluate the same scenarios with additional restrictions on air emissions and discuss our findings.

A STRUCTURAL-DYNAMIC MODEL OF A DAIRY FARM

Figure 1 summarizes the key inputs and outputs (bold text), choice variables (ovals), and intermediate calculations (gray boxes) in our modeling approach. Due to the level of detail, much of the model exposition is contained in an appendix available from the authors upon request. This includes parameter values and functional forms for the *herd management*, *waste management*, and *crop production components*. In the main text we present the important variables and relationships that are necessary for understanding our general approach.

Herd Management

Our model farmer works in discrete time and manages a self-replacing herd of calves, heifers, and milk cows. Each year the farmer decides how many animals from each age cohort to retain and how many to sell (cull), and how many replacement heifers to purchase. The equations of motion for the cohorts can be expressed as a vector function \mathbf{H}

$$\mathbf{h}_{t+1} \equiv \mathbf{H}(\mathbf{h}_t, \boldsymbol{\theta}_t, \omega_t, \boldsymbol{\gamma}^h) \quad (1)$$

where \mathbf{h}_t is a vector representing the number of animals in each cohort during year (t); $\boldsymbol{\theta}_t$ is a vector representing the culling rates; ω_t is the number of replacement heifers purchased; and $\boldsymbol{\gamma}^h$ is a parameter vector describing herd characteristics such as birth and mortality rates.

Dairy farmers control their aggregate milk, meat, and waste outputs by varying both the herd size and the inputs provided to each cow. In reality, determining the optimal combination of inputs is quite complicated. Rotz et al (1999) list 30 different constituents that may be used by farmers to develop a ration. These constituents exhibit fluctuating availabilities, prices, and qualities; they are marked by complicated patterns of substitutability; and they are bounded by multiple constraints. To simplify this aspect of the problem, we follow convention and assume each milk cow consumes a fixed cohort-specific ration. Furthermore, because the marginal contributions of each input to milk, meat, and waste outputs are largely unknown, we also assume that each cow achieves a cohort-specific weight (used to determine the cull price) and produces a fixed amount of milk and waste during each lactation. With this specification, our herd model exhibits constant returns to scale. However, as is common for modern dairies, we also include a herd permit constraint that limits the total number of animal units.

Given the preceding, we can write the herd component of the profit function as

$$\pi_t^h \equiv \Pi^h(\mathbf{p}^h, \mathbf{x}^h, \mathbf{h}_t, \boldsymbol{\theta}_t, \omega_t, \boldsymbol{\gamma}^h) \quad (2)$$

where \mathbf{p}^h is a vector of input and output prices; \mathbf{x}^h is a vector of fixed per-cow inputs and outputs; and the other variables are defined previously.

Waste Management

The second major component of the dairy operation is waste handling and disposal. The amount and composition of waste can vary substantially across farms, depending on the type of housing (e.g., free stall, corral, open lot), manure collection system

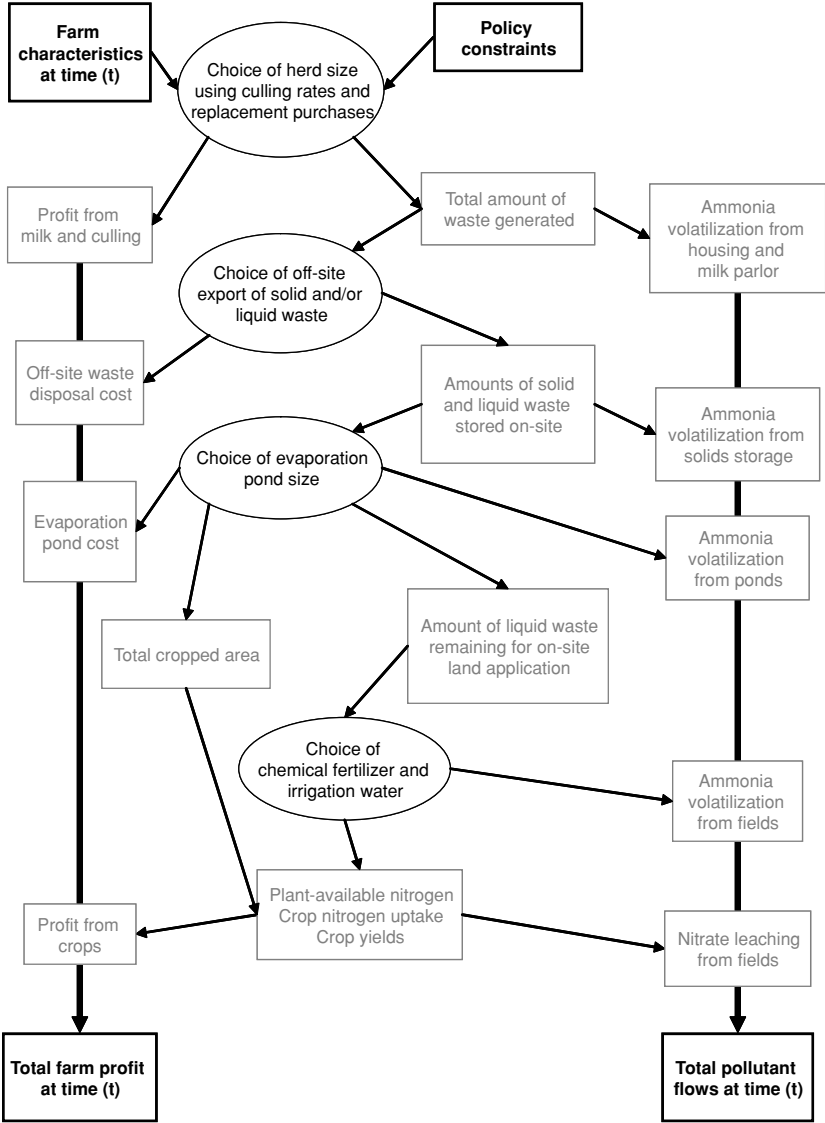


Figure 1. Key model inputs, choice variables, intermediate calculations, and outputs

(e.g., flush, scrape, vacuum), waste treatment (e.g., solids screening, composting, aerobic/anaerobic digestion), waste storage (e.g., lagoons, tanks, stacks), and environmental conditions (e.g., climate). In California's San Joaquin Valley and elsewhere, it is common for large modern dairies to employ free stall housing with waste flushing, solids screening, lagoon storage of liquids, and stacking of dried solids. Solid and liquid wastes are deposited in both the housing structure and the milking parlor and then flushed with water into a solids separator that removes a fraction of the solid content. The separated

solids are dried and placed in a manure storage facility; the liquids are stored in an open lagoon. As this is a typical process for modern dairies and because we have excellent data from a farm like this near Hilmar, California,⁴ we specify this type of waste handling system for our model and leave an investigation of alternative systems for future work.

Even with these specifications, the characteristics of the final waste product depend on numerous decisions made by the farmer, including: the quantity and quality of flush water; the flushing frequency; the amount and type of bedding material used; and—because nitrogen is not a conservative pollutant—the residence times in various stages of the waste handling system. Following convention, we assume the farmer cannot affect aspects of the waste handling system that occur between waste generation and storage. Rather, for a given quantity of generated waste (which the farmer affects through herd management decisions), the resulting flows to solid and liquid storage are predetermined; the farmer then determines how to dispose the stored waste.

Due to differing transportation costs and marketable end-uses, large dairies often sell dried solid waste but retain liquid waste for irrigating and fertilizing crops. However, NMPs will require farmers to significantly reduce their onsite application rates. The literature cited above suggests that farmers are likely to change their waste management practices by (1) reducing the quantity of stored waste by increasing the ammonia volatilization rate and (2) exporting additional stored waste by paying a custom applicator to haul liquid manure to nearby cropland. We incorporate the first response into our model by allowing the farmer to implement evaporation ponds. We do this for several reasons. First, evaporative disposal already is used by some California dairies (Morse-Meyer et al 1997). Second, although nitrogen emissions to ground water and air historically have been treated as separate problems,⁵ each is a result of the same waste stream generated by the milking herd. Therefore, when faced with regulations on emissions into one medium, a farmer naturally would attempt to take advantage of the remaining free disposal option before undertaking costly pollution control measures (NRC 2002; Aillery et al 2005). And third, although there may be other ways to increase ammonia volatilization from a dairy, we note that evaporation of saline drainage water is a well-established, cost-effective waste disposal practice for crop producers in arid and semiarid regions.

We incorporate the second response by specifying an off-site waste disposal cost function that depends on the quantity of exported waste and the distance hauled. Following convention, we assume distance is a function of the suitability and capacity of nearby land for receiving manure nutrients as well as the willingness of the land owners to accept waste. To simplify the dynamics of our problem we assume no waste is carried over between crop seasons, implying all waste generated during each season must volatilize, be land applied, or be exported off the farm during that season.

Given the preceding, we can incorporate the revenue from dried solid waste, the cost to haul and apply liquid waste, and the cost to install and maintain additional lagoon surface area into a single waste disposal cost function

$$\pi_t^d \equiv \Pi^d(l_{ct}, s_{ct}, e_t, \mathbf{p}^d, \boldsymbol{\gamma}^d) \quad (3)$$

where l_{ct} and s_{ct} are the amounts of liquid and solid wastes applied at the dairy; e_t is the total surface area of the lagoons; \mathbf{p}^d is a vector of unit costs; and $\boldsymbol{\gamma}^d$ is a parameter vector

including information about the characteristics of the stored waste and the receiving land.

Crop Production

The third and final component of the dairy farm is crop production. Here we follow convention and assume farmers grow two crops annually—summer corn and winter wheat—on a fixed amount of land that is available for either crop production or waste lagoons. A notable aspect of this model component is the nonuniformity of the irrigation system, which has been shown to significantly affect soil nitrogen levels and nitrate leaching rates (Knapp and Schwabe 2008) but, which has been absent from previous studies of livestock-crop operations.⁶ Nonuniform irrigation is modeled by a parameter $\beta \in [0, \infty]$, which represents the water infiltration coefficient (i.e., the fraction of applied water that infiltrates into the root zone) at each point in the field and which has a log-normal distribution $g(\beta)$ per unit area. We can therefore specify the equations of motion for the soil nitrogen concentrations at any point in the field as a vector function \mathbf{N}

$$\mathbf{n}_{ct+1}(\beta) \equiv \mathbf{N}(\mathbf{n}_{ct}(\beta), s_{ct}, l_{ct}, f_{ct}, i_{ct}, \boldsymbol{\gamma}^n) \quad (4)$$

where $\mathbf{n}_{ct}(\beta)$ is a vector of organic and inorganic soil nitrogen concentrations; s_{ct} , l_{ct} , f_{ct} , and i_{ct} are control variables representing the amounts of solid and liquid waste, commercial fertilizer, and irrigation water applied to fields; and $\boldsymbol{\gamma}^n$ is a parameter vector.⁷ Applications of liquid waste also are subject to a constraint that they must be sufficiently diluted with irrigation water to avoid damaging crops with high concentrations of waste components that do not volatilize (e.g., salts) and therefore become concentrated in the residual lagoon water (Swenson 2004).

Crop production at any point in the field can be expressed similarly as a function Y

$$y_{ct}(\beta) \equiv Y(\mathbf{n}_{ct}(\beta), s_{ct}, l_{ct}, f_{ct}, i_{ct}, \boldsymbol{\gamma}^y) \quad (5)$$

where $\boldsymbol{\gamma}^y$ is a parameter vector. Nitrogen leaching and ammonia volatilization from any point in the field also can be expressed as functions of the same state and control variables. Aggregate crop yields are calculated by integrating Y over $g(\beta)$ and multiplying by the total cropped area; aggregate amounts of leaching and volatilization from fields are calculated by similarly integrating over equations describing leaching and volatilization rates at each location.⁸

Given the preceding, we can write each crop component of the profit function as

$$\pi_{ct}^y \equiv \Pi^y(\mathbf{p}^y, \mathbf{x}^y, \mathbf{n}_{ct}(\beta), s_{ct}, l_{ct}, f_{ct}, i_{ct}, e_t, \boldsymbol{\gamma}^n, \boldsymbol{\gamma}^y) \quad (6)$$

where \mathbf{p}^y is a vector of input and output prices; \mathbf{x}^y is a vector of fixed inputs to the cropping system; and the other variables have been defined previously (with $\boldsymbol{\gamma}^y$ now also including the parameters describing the distribution g).

Optimization

Defining $\pi_t \equiv \pi_t^h + \sum_c \pi_{ct}^y - \pi_t^d$, collecting all prices into a vector \mathbf{p} and all parameters into a vector $\boldsymbol{\Gamma}$, specifying a discount factor ρ and a time horizon T , and assuming

farmers maximize the net present value (NPV) of farm operations, we can summarize the essential components of the producer's problem as

$$\max_{\{\theta_t, s_{ct}, l_{ct}, f_{ct}, i_{ct}, \omega_t, e_t\}} \left[\sum_{t=0}^T \rho^t \pi_t(\mathbf{h}_t, \mathbf{n}_{ct}(\beta), \theta_t, s_{ct}, l_{ct}, f_{ct}, i_{ct}, \omega_t, e_t \mid \mathbf{p}, \mathbf{\Gamma}) \right] \quad (7)$$

subject to the equations of motion for the herd and the soil nitrogen concentrations, constraints on total available land and total allowable animal units, mass balance constraints on solid and liquid waste streams, and the liquid waste dilution constraint. This statement defines an optimal control problem with state variables for the herd age cohorts and soil nitrogen concentrations, and with control variables for the culling rates, the application rates for solid waste, liquid waste, chemical fertilizer, and irrigation water, the number of purchased replacement heifers, and the evaporation pond area. We solve this dynamic optimization problem in GAMS as a constrained nonlinear programming problem (Standiford and Howitt 1992) utilizing the CONOPT solver.

Our first goal is to find a dynamic steady state and verify that our model farm is representative of our study site in Hilmar, California; then we conduct policy simulations and sensitivity analyses. To find feasible starting values for the steady-state search, we first treat the model as a period-by-period optimization problem: we choose a set of initial conditions, optimize the first period in isolation from the others, use the state equations to “roll forward” to the next period, and continue until the last period (which is set large enough to avoid boundary effects). We then solve the dynamic problem using the period-by-period solution as the starting values, check if the model has reached a steady state, select a new set of initial conditions from the dynamic solution path, and repeat until steady-state convergence criteria are satisfied.

MODEL CALIBRATION RESULTS

Table 1 summarizes the results of our model calibration by comparing various steady-state values against available data. Despite the large number of parameters, variables, and equations, and the complexity of the optimization problem, the model appears to be calibrated well. Animal cohort numbers are similar to those reported by VanderSchans (2001) for the Hilmar site. Differences are most likely due to off-farm rearing of some calves and heifers (a strategy which is not chosen by our model farm). Income data were not available for the Hilmar farm, but we can compare our annual profit per cow against Rotz et al (2003) who simulate a 1,000 cow dairy with 770 heifers and 600 hectares (ha) of cropland. Our profit per cow is low compared to their estimate, but this appears to be due to different assumptions about milk yield. The average annual milk yield for our herd is 9,509 kg/cow whereas the average for the simulation in Rotz et al (2003) is 11,300 kg/cow. Substituting 11,300 kg/cow into our model gives annual profit of \$1,239/cow, which is close to their estimate. However, we retain the lower values because they are much closer to the reported average for California dairy cows (USDA 2006b).

Ammonia volatilization from our model farm is similar to reported values, and nitrate leaching is nearly identical to VanderSchans' best estimate (based on a hydrologic model) for the Hilmar farm. Corn and wheat yields are high but within reason, as are the concentrations of nitrogen in the manure storage lagoon (all of which are compared to

Table 1. Model calibration results

Quantity	Units	Steady-state value	Comparison value	Comparison source ^a
Calves	No. of animals	723	517	VanderSchans 2001
Heifers	No. of animals	577	308	VanderSchans 2001
Milk cows	No. of animals	1445	1731	VanderSchans 2001
Replacement heifers purchased	No. of animals	0	—	—
Annualized profit per milk cow (\$2005)	\$/head	706	1309	Rotz et al 2003
Ammonia volatilization	kg N/head-yr	41 ^b	38	USEPA 2004
			64	Chang et al 2004
Nitrate leaching	kg N/ha-yr	414	417	VanderSchans 2001
Corn yield	T/ha-yr	10.8	6.7–13.3	Vargas et al 2003
			7.2–10.0	Crohn 1996
Wheat yield	T/ha-yr	7.9	4.2–6.7	Brittan et al 2004
			2.7–7.7	Crohn 1996
Lagoon nitrogen concentration	mg N/l	895	200–1000	VanderSchans 2001
Lagoon inorganic nitrogen concentration	mg N/l		500–800	Campbell Mathews 2006
Applied water (irrigation + pond)	cm/yr	395	300–600	Chang et al 2005
Applied chemical (nitrogen) fertilizer	kg N/ha-yr	111	124	VanderSchans 2001
Applied solid manure	kg N/ha-yr	0	130–280	VanderSchans 2001
		0	—	—

^a VanderSchans 2001 corresponds to comparison values from the Hilmar farm.
^b Includes heifers and milk cows but not calves. Annual volatilization per milk cow is 57 kg N.

other published sources due to lack of data for the Hilmar farm). Applied water (irrigation plus lagoon water) is close to the Hilmar farm estimate, but applied chemical (nitrogen) fertilizer is significantly different. Our model farm does not apply any chemical fertilizer, which supports results by Chang et al (2005) that California dairies can achieve high crop yields without chemical fertilizers; but it contradicts observed practice at the Hilmar site. However, the only noteworthy changes derived from imposing the midpoint application rate of 205 kg N/ha-yr on our model is a 1% decrease in profit and a 7% increase in the leaching rate. Last, and consistent with VanderSchans (2001), our model farm sells and exports all dried solid manure.

NUTRIENT MANAGEMENT PLAN SIMULATIONS

NMPs are readily incorporated into our modeling framework by specifying an additional constraint that limits the amount of nitrogen that may be land applied each year at the dairy. Following convention, the land application constraint is set equal to the estimated total amount of nitrogen contained in the harvested portions of the cropping system, plus an allowance for unavoidable soil nitrogen losses. To make our constraint consistent with previous studies, quantities of harvested nitrogen are based on crop-specific nutrient uptake rates published by Lander et al (1998), and the allowance for unavoidable losses is taken from Kellogg et al (2000). This gives a maximum nitrogen application rate of 412 kg N/ha-yr, whereas the total amount of applied nitrogen in the unregulated steady state is 2,196 kg N/ha-yr.

Our policy simulations assume the dairy farm is initially at the steady-state operating position derived in the model calibration section. We then introduce the NMP constraint and derive the dynamically optimal response for the dairy. We focus on the change in the NPV of farm operations during the simulated time period, as well as the time paths for three variables: herd size [number of milk cows], nitrate leaching [kg N/ha-yr], and ammonia volatilization [kg N/yr]. Again following convention, we present the results for different levels of “willingness to accept manure” (WTAM) by surrounding land operators. WTAM is the percentage of surrounding land suitable for receiving manure that is willing to accept it. For our study site, we calculate that 25% of surrounding land is suitable for receiving manure (Kellogg et al 2000; USDA 2006c); the WTAM values we consider therefore correspond to 25%, 15%, 5%, and 1% of surrounding land (e.g., in Table 2: 4% of 25% = 1%).⁹

Scenario 1 in Table 2 shows the policy-induced NPV loss and new steady-state levels for the other variables of concern given our baseline model parameter values. The predicted loss ranges from 12 to 19% of NPV, depending on WTAM. Previous estimates for implementing nitrogen-based NMPs at “large” dairy operations (typically ≥ 700 cows) are in the range of 2–6% of profits (Ribaud et al 2003; Aillery et al 2005; Huang et al 2005; Ribaud and Agapoff 2005). Whereas these studies focus on off-site manure disposal, our estimate includes 5–12% from off-site disposal of liquid waste (Table 2 shows average hauling distances), 4.7% from efforts to increase ammonia volatilization (total pond area increases from 1.1 ha prepolicy to 11.0 ha postpolicy), and 2.3% from reduced production (lower crop yields due to less applied water and nitrogen and less cropped area due to displacement by evaporation ponds). Although this result confirms that off-site disposal will be a key response to NMP requirements, it does not support the notion that

Table 2. Steady-state NMP simulation results without air regulations for various model scenarios and various levels of willingness to accept manure

WTAM	NPV loss [%]	Milk cows [#]	Applied water (irrigation + pond) [cm/yr]	Evaporation pond area [ha]	Average waste hauling distance [km]	Leaching [kg N/ha-yr]	Volatilization [kg N/yr]
Scenario 1: Baseline parameter values with 412 kg N/ha-yr application limit							
100%	12.3	1,445	55.7	11.0	1.8	6.0	130,569
60%	12.7	1,445	55.7	11.0	2.3	6.0	130,569
20%	14.3	1,445	55.7	11.0	4.0	6.0	130,569
4%	18.8	1,445	55.7	11.0	8.9	6.0	130,568
Scenario 2: Improved input management with 412 kg N/ha-yr application limit							
100%	10.3	1,445	61.6	11.0	1.1	8.6	65,834
60%	10.5	1,445	61.6	11.0	1.4	8.6	65,834
20%	11.0	1,445	61.6	11.0	2.3	8.6	65,834
4%	12.7	1,445	61.6	11.0	5.1	8.6	65,834
Scenario 3: Uniform irrigation with 1,200 kg N/ha-yr application limit							
100%	6.2	1,445	69.9	11.0	1.1	6.0	132,670
60%	6.3	1,445	69.9	11.0	1.4	6.0	132,670
20%	6.6	1,445	69.9	11.0	2.4	6.0	132,670
4%	7.5	1,445	69.9	11.0	5.3	6.0	132,669
Scenario 4: Selective culling with 412 kg N/ha-yr application limit							
100%	10.6	1,392	55.7	10.6	1.8	6.0	130,044
60%	11.0	1,392	55.7	10.6	2.3	6.0	130,044
20%	12.3	1,392	55.7	10.6	4.0	6.0	130,044
4%	16.1	1,392	55.7	10.6	8.8	6.0	130,044

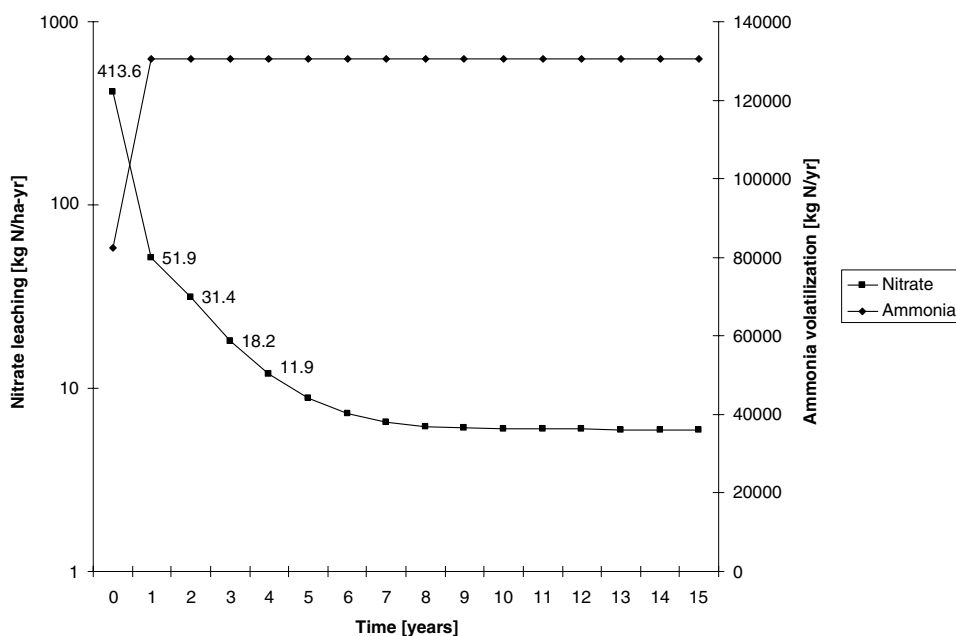


Figure 2. Time paths for nitrate leaching and ammonia volatilization for the baseline scenario without air regulations

a simpler analysis focusing on waste disposal costs alone will be sufficient for estimating the economic implications for producers. We revisit this finding and discuss additional implications in the concluding section.

The other variables in Table 2, which characterize the new steady-state operating position of the dairy, are not affected by WTAM in this scenario. Relative to the unregulated steady state, the herd size remains unchanged at 1,445 milk cows, the leaching rate falls from 413 to 6 kg N/ha-yr, and the volatilization rate increases from 82,463 to 130,569 kg N/yr. Figure 2 shows that the leaching rate falls precipitously during the first year and then much more gradually thereafter (note the logarithmic scale). After four years the leaching rate is still twice as high as the eventual steady-state value, but after eight years it is within 10% of this value. These results are consistent with the literature on nitrate leaching from crop operations (Knapp and Schwabe 2008) and, together with the result for the herd size, suggest that the dynamics of NMP implementation in this scenario are primarily captured by the crop production component of the model rather than the herd component. However, we will see that culling decisions play a more prominent role when NMPs are implemented in conjunction with ammonia regulations.

Finally, we observe a 58% increase in volatilization of ammonia emissions for this scenario. The increase in ammonia emissions is substantially larger than the only comparable estimate we can find elsewhere (for hog operations, by Aillery et al 2005), and is likely due to the additional control variable in our model, which allows the farmer to increase lagoon surface area. Apparently, this is a low-cost response to NMP requirements that can produce a significant increase in ammonia emissions; in fact, our model predicts that

farmers will maximize lagoon emissions for all values of WTAM. Figure 2 shows that the time path of ammonia emissions is qualitatively similar to that for nitrate leaching: the new steady-state value is attained during the first year of NMP implementation with no additional increases thereafter.

SENSITIVITY ANALYSIS

Similar to previous studies, the preceding analysis does not account for the possibility that, when faced with new waste disposal restrictions, farmers may attempt to implement (currently unproven) input management practices in an effort to reduce costs.¹⁰ For example, research suggests that the nitrogen concentration of the waste stream may be reduced 20–40% by feeding amino acid supplements (Kohn 1999), 8–15% by grouping and feeding cows according to milk production levels (Castillo 2003), and nearly 10% by adjusting the composition of the feed ration (Jonker et al 2002). Dunlap et al (2000) estimate that feeding bovine growth hormone, milking three times daily, and exposing cows to artificial daylight during nighttime collectively can reduce waste nitrogen by 16%. To the extent these practices are currently used by California dairies, our model implicitly accounts for their impacts on milk production and waste generation because we calibrate our model with statewide per-cow averages. Assuming none is widely used, the nutrient content of the waste stream could be approximately halved if all of these practices were implemented. However, a significant (and still largely unknown) cost would be incurred either by the farmer or by an agency offering adoption subsidies for these practices. To conduct a sensitivity analysis and to assess the potential benefit of these techniques, we assume our model farm adopts all of these fully subsidized practices (i.e., at no cost) and achieves a 50% reduction in the nitrogen concentration of the waste stream.

Scenario 2 of Table 2 presents these policy simulation results. Relative to scenario 1, adopting these practices saves the farmer 2–6% of net income depending on WTAM. Whether or not these gains would offset adoption costs in the absence of government subsidies is a question we currently cannot answer; here we consider the effect on steady-state nitrogen emissions. Relative to the baseline policy simulations, halving the nitrogen concentration of the waste stream reduces ammonia emissions by 49% but increases nitrate leaching from 6.0 to 8.6 kg N/ha-yr.¹¹ The increased leaching arises from multiple effects. First, with a lower nitrogen concentration in the waste, more waste is retained on the farm for land application. Second, because this waste contains the same concentration of salts as it did in the baseline case, relatively more irrigation water (about 10%) must be applied to achieve sufficient dilution. This additional water flushes more nitrates through the soil and increases the leaching rate.

This counterintuitive result suggests that the problem of nitrogen emissions should not be considered as a simple nutrient mass-balance problem, but rather as a more complicated problem involving relationships between nutrients, water, and other waste components.¹² It also suggests that improved irrigation uniformity could allow the NMP constraint to be relaxed without increasing the leaching rate because less water would pass through the rootzone and into the aquifer. In fact, assuming perfectly uniform irrigation, our model predicts that the NMP constraint could be increased from 412 to 1,200 kg N/ha-yr while still achieving 6 kg N/ha-yr of nitrate leaching. This is largely due to increased denitrification (conversion of nitrate to benign dinitrogen gas) and crop uptake,

but also entails higher ammonia emissions from fields. The associated NPV loss would be reduced to 6–8% of net income, depending on WTAM, without any improvements to input management. These results are summarized as the third scenario in Table 2; policy implications are discussed later.

Another management alternative overlooked by the existing literature (and our baseline scenario) is that of selectively culling lower producing animals when faced with waste disposal restrictions, which also would tend to reduce NMP implementation costs relative to the case of homogenous age cohorts. Although culling models do exist (e.g., Van Arendonk 1985), they have not been used in the context of environmental pollution control. We use our model to approximate such culling decisions by introducing cohort-specific milk yield distributions and assuming farmers cull the lowest yielding cows first. Specifically, we assume each cohort milk yield distribution is uniform with mean given by the cohort-specific milk yield used in the baseline scenario and with the highest yielding cow producing twice as much as the lowest yielding cow.¹³ This gives a slightly different unregulated steady-state operating position for the farm: profits are 13% higher, the herd contains 1,392 milk cows, leaching is 404 kg N/ha-yr, and volatilization is 82,358 kg N/yr. Scenario 4 of Table 2 presents the policy simulation results relative to these unregulated steady-state values. The response of the dairy for all WTAM values is similar to the response in scenario 1, which assumed a homogenous herd: the herd size remains unchanged, leaching drops substantially, and volatilization increases by 58%. Interestingly, the ability to cull low yielding cows reduces the percentage income loss by only 2–3% relative to scenario 1, suggesting that such decisions may not play a major role in NMP implementation.

NMP SIMULATIONS WITH AIR REGULATIONS

Given our predictions of substantial policy-induced increases in ammonia volatilization and the documented air quality problems in livestock-intensive regions, we now consider the likely effects of implementing ammonia regulations in addition to NMPs. Regulations on ammonia emissions could take a variety of forms; as in Aillery et al (2005), we consider the relatively straightforward case of a quantity restriction. The regulation we consider requires that total ammonia emissions from the farm not exceed the unregulated steady-state level. This may be a relatively lenient restriction, given that air quality regulators in California are pursuing strategies to reduce ammonia emissions from AFOs.

Policy simulation results for the same scenarios considered above are given in Table 3. The second scenario (improved input management) is identical to that of Table 2 because the optimal strategy for this scenario without air regulations is to reduce volatilization below the unregulated steady-state value; therefore the additional air quality regulation is not binding. However, the results for the other scenarios are significantly different from those in Table 2. For the baseline parameter values the expected loss is now much higher at 37–45% of net farm income, depending on WTAM. These estimated losses are about 2–3 times as high as the most comparable estimates in the existing literature (Aillery et al 2005). With restrictions on both waste streams, Table 3 shows it is now optimal to reduce the herd size and incur both crop and livestock production losses in scenarios 1, 3, and 4. Other management responses are qualitatively similar to those without air regulations: applied water and nitrogen are reduced to similar levels, liquid waste is shipped off-site,

Table 3. Steady-state NMP simulation results with air regulations for various model scenarios and various levels of willingness to accept manure

WTAM	NPV loss [%]	Milk cows [#]	Applied water (irrigation + pond) [cm/yr]	Evaporation pond area [ha]	Average waste hauling distance [km]	Leaching [kg N/ha-yr]	Volatilization [kg N/yr]
			Scenario 1: Baseline parameter values with 412 kg N/ha-yr application limit				
100%	37.6	1,212	55.7	5.8	1.9	5.1	82,463
60%	38.9	1,150	55.7	6.0	2.3	5.2	82,463
20%	41.8	1,036	55.7	6.3	3.4	5.5	82,463
4%	45.1	913	55.7	6.5	6.2	5.9	82,463
			Scenario 2: Improved input management with 412 kg N/ha-yr application limit				
100%	10.3	1,445	61.6	11.0	1.1	8.6	65,834
60%	10.5	1,445	61.6	11.0	1.4	8.6	65,834
20%	11.0	1,445	61.6	11.0	2.3	8.6	65,834
4%	12.7	1,445	61.6	11.0	5.1	8.6	65,834
			Scenario 3: Uniform irrigation with 1,200 kg N/ha-yr application limit				
100%	22.1	1,409	68.3	3.3	1.6	4.3	82,463
60%	23.3	1,352	68.5	3.8	1.9	4.4	82,463
20%	26.0	1,241	68.4	4.7	2.8	4.5	82,463
4%	29.1	1,119	68.3	5.4	4.4	4.6	82,463
			Scenario 4: Selective culling with 412 kg N/ha-yr application limit				
100%	33.4	1,229	55.6	5.3	2.0	5.0	82,358
60%	35.0	1,163	55.7	5.6	2.4	5.1	82,358
20%	38.3	1,035	55.7	6.0	3.6	5.4	82,358
4%	42.0	896	55.8	6.3	6.4	5.9	82,358

and evaporation ponds are implemented—although to a lesser extent due to restrictions on air emissions—thus displacing cropped area. Though not shown graphically, herd reductions are qualitatively similar to nitrate leaching reductions: large reductions occur during the first one to two years, followed by smaller reductions (and sometimes small cyclical fluctuations) thereafter. In scenarios 1 and 4 the associated production losses represent a large portion of the total loss: 15–35% of net farm income depending on WTAM. In scenario 3 these production losses range from 3–21% of the total. Selective culling again does not have a large effect on costs, and improved irrigation uniformity has a relatively smaller effect than it does in the absence of air regulations.

DISCUSSION AND CONCLUSIONS

Economies of scale and technological innovation are resulting in more concentrated animal feeding operations worldwide. Governments are reacting to the associated waste management problem primarily with tighter restrictions on nutrient application rates to protect water quality. However, a potentially perverse outcome from these more stringent nutrient restrictions is an incentive to increase volatilization of nitrogen, often in areas located near population centers and/or in areas where air quality already is degraded (FAO 2007).

This study focuses on the dairy industry, which increasingly has been the target of NMPs in the European Union, New Zealand, Canada, and the United States. We develop a structural-dynamic model of a modern dairy farm, including milk and livestock production, waste generation, treatment, and disposal, and crop production with nonuniform irrigation. The model is calibrated with farm-level data from a well-documented dairy in the San Joaquin Valley and with additional data from other sources. The optimized characteristics of the farm, including herd size, crop yields, amounts of applied water, nitrate leaching, ammonia volatilization, and net farm income are consistent with available comparison data.

Regarding our first objective—to estimate producer costs with a detailed structural-dynamic model—we find that implementing nitrogen-based NMPs could generate profit losses around 12–19%, substantially greater than the most comparable estimates from previous studies. Unfortunately, our modeling approach is sufficiently different from those studies such that those models cannot be characterized as constrained versions of our own, thus making it difficult to isolate the exact sources of the cost differences. We suspect that some of the difference is inherent in the modeling structure itself (just as alternative models of recreation behavior, for example, will generate different estimates of recreation value from the same data set), particularly because our model includes an additional disposal option—increased pond volatilization—which intuitively should decrease compliance costs. The relatively severe reduction in land application of nitrogen required by the NMP constraint also would tend to produce higher compliance costs; however the reduction is calculated using conventional methods and the economic loss is reported as a percentage to help control for farm size. Another likely source of the cost difference is the characteristics of our simulated farm: we have chosen to model a dairy farm that uses modern technologies, operates in a semiarid environment, and is relatively large by comparison to “large” farms that have been modeled in previous studies. We do this in light of the direction the industry is taking toward consolidation, particularly

in areas, such as the western United States, where both groundwater and air quality problems persist. We also think that the stakes are highest for such farms because they produce a large share of total output and therefore their operating decisions could have nontrivial effects on local economies, markets, and even trade (Cassells and Meister 2001). Expected losses around 12–19% could make policy implementation difficult in some regions or induce unanticipated changes in the industry (e.g., restructuring, relocation) as major producers adjust to the regulations. Overall, we think NMP implementation will have a greater economic impact on large producers than has been suggested by previous farm-level studies.

In terms of how these costs might be reduced, our simulations suggest two promising avenues: improved input management and irrigation uniformity. According to our estimates in Table 3, improved input management has the potential to reduce economic losses by 75% when both NMPs and air regulations are implemented together. However, this finding is based on assumptions about currently unproven technologies and the costs producers might incur to adopt them. It also comes with the caveat that nitrate leaching may actually increase slightly as the nitrogen throughput of an AFO decreases; but this observation simply reinforces our belief that regulating the application of nitrogen alone is not the best approach to the problem. Regardless, research is needed to develop these technologies, identify their cost functions, and examine what types of additional incentives, if any, might be appropriate for encouraging their use. Similarly, our analysis of uniform irrigation also demonstrates the potential benefit of this management option but does not investigate the costs to adopt alternative irrigation systems. Because irrigation cost data are much more readily available, modeling endogenous irrigation system choice would be a logical extension of this research.

Improved irrigation uniformity also could reduce implementation costs if NMP restrictions are relaxed accordingly; however, to our knowledge such allowances currently are not being considered. By regulating nitrogen application rates rather than leaching rates, regulators are missing an opportunity to encourage producers to adopt less polluting and potentially cost-saving irrigation systems. This is a case of regulating a precursor to pollution rather than the pollution itself, which typically produces an inefficient outcome. An incentive could be created, for example, if the NMP constraint were related to the irrigation system choice such that users of more uniform systems were allowed to apply more nitrogen.

Regarding our second objective—to estimate NMP-induced environmental impacts, in particular the time required for reductions to be achieved and the potential for cross-media effects—we find that initial reductions in nitrate leaching will occur quickly but achieving steady-state levels will require seven to nine years. We also predict that ammonia emissions will increase rapidly and there is considerable risk of substantially degrading air quality if NMPs are implemented without ammonia regulations. These results differ from recent work by Aillery et al (2005) who find a notably smaller potential for cross-media pollution from hog operations. Our results suggest more research is needed to better understand the potential environmental and economic tradeoffs associated with AFO regulations and to determine what should be done to manage these tradeoffs appropriately. Issues to consider include the benefits obtained from reducing emissions, including the temporal aspect of exposure to both nitrate and ammonia: whereas ammonia emissions can have an immediate effect on air quality, nitrate emissions may take longer to migrate

through the hydrologic system before impacting a recreational resource or a drinking water source. Such an analysis also should consider that ammonia alone does not create airborne particulate matter but rather must interact with sulfur or nitrogen oxides, which primarily are the result of combustion processes. Given the high cost we estimate to implement both water and air regulations, increased ammonia emissions may be deemed acceptable in regions that are oxide-limited. Population and climate variables also will affect this tradeoff, and it is likely that populous arid regions that rely on ground water resources will face the most difficult choices.

Last, regarding our third objective—to advance the modeling techniques used to predict the effects of environmental regulations on AFOs and to evaluate whether the additional model detail and effort produce significantly different outcomes—we find somewhat mixed results. On the one hand, the differences between our results and those of previous studies, as well as the additional temporal insights generated by a multiperiod framework, suggest that structural-dynamic modeling of AFO regulation should not be dismissed as “not worth the trouble.” More work is needed to clarify the exact sources of the differences and to determine if other potentially important aspects of the problem (i.e., the waste dilution constraint, irrigation system uniformity) have been overlooked. A formal comparative modeling analysis is beyond the scope of this work but would be a useful next step. On the other hand, we find that herd management dynamics are not as important as soil nitrogen dynamics in much of the present analysis. Most likely this is because each age cohort can be controlled (culled) separately, which effectively relaxes the constraints imposed by the state equations and makes the herd management component behave more like a static optimization problem. A simpler approach that still includes soil nitrogen dynamics but omits the formal state equations for the herd age cohorts while still allowing the operator to choose a herd size might be an appropriate compromise between fully static and dynamic models.

NOTES

¹It is noteworthy that NMPs regulate a management practice rather than emissions and therefore this approach is unlikely to be cost-effective because it fails to provide an incentive for farmers to seek out the least-cost combination of all available management practices. We revisit this point later with regard to irrigation system choice.

²Previous studies have incorporated dynamic elements when examining livestock management decisions (e.g., Van Arendonk 1985; Chavas and Klemme 1986; Tozer and Huffaker 1999) but not in the context of environmental regulation. The only dynamic analysis of livestock production and environmental regulation that we are aware of is Schnitkey and Miranda (1993). Other studies (e.g., Kim, et al 1993; Yadav 1997; Nkonya and Featherstone 2000) have demonstrated the importance of dynamic elements affecting the fate and transport of nitrates in the environment.

³California's San Joaquin Valley provides an appropriate test bed for our analysis. California is home to nearly 20% of all U.S. dairy cows and produces 21% of the nation's milk, primarily from relatively large farms (USDA 2006a). Although runoff pollution from California dairies is well-regulated, between 10 and 15% of the State's water supply wells exceed the federal standard for nitrate largely due to ground water infiltration from agricultural fields (Bianchi and Harter 2002). The San Joaquin Valley Air District, home to numerous dairy operations, currently violates federal standards for particulate matter air pollution with 43% of ammonia emissions—a precursor to particulate pollution—originating from dairies (Palsgaard 2006).

⁴VanderSchans (2001) provides a detailed description of the study farm.

⁵For example, many of the manure management strategies suggested by the Dairy Permitting Advisory Group for the San Joaquin Valley Air Pollution Control District involve shifting emissions from ammonia to nitrate (Abernathy et al 2006).

⁶Nonuniform irrigation refers to the fact that when, for example, a 1 ha field is irrigated with 10 cm of water, some areas necessarily receive more than 10 cm of water and others receive less (Vickner et al 1998; Anselin et al 2004; Knapp and Schwabe 2008). Nonuniform irrigation leads to higher leaching rates because producers find it optimal to apply more than 10 cm-ha of water to ensure the relatively dry areas are adequately irrigated, with excess water in wet areas carrying pollutants down into the aquifer.

⁷Here we use the subscript $ct + 1$ as shorthand notation for the next cropping season, which could be either the next season of the same year or the first season of the next year.

⁸Details are provided in the appendix available from the authors upon request. Pond volatilization is based on a standard physical relationship that accounts for the aqueous ammonia nitrogen concentration, climatic conditions, and total pond surface area (Liang et al 2002).

⁹Our study site is located in an area where off-site disposal of manure should be relatively cheap. A relatively large share of the surrounding land is intensively farmed and able to receive substantial quantities of waste nitrogen (Kellogg et al 2000). Therefore the NMP implementation cost for our model farm will tend to be less than for a similar farm facing competition for land from other AFOs, high-value agricultural producers, or urban developers. Furthermore, because we use straight-line distances to calculate hauling costs, our disposal cost estimates will tend to be less than those for an actual dairy.

¹⁰De Vos et al (2003) provide thorough analysis of how alternative manure management systems affect waste residuals and returns for hog operations.

¹¹Johansson and Kaplan (2004) derive a similar result using a different modeling approach.

¹²The observation that water application rates are an important component of the nitrate leaching problem is consistent with the findings of Knapp and Schwabe (2008).

¹³Available data on within-herd milk yield variability is limited. Cassel (2001) reports that one rating system classifies cows into five groups, with the highest producing at least 110% of the herd average and the lowest ("probable cull cows") producing less than 80% of the average. Several sources (e.g., Wattiaux 2003) suggest the distribution is approximately normal. Our assumptions therefore are optimistic: the variability is somewhat larger than in Cassel (2001) and there are relatively more cows in the tails of the distribution which translates into larger potential efficiency gains from culling.

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